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Rangeland Ecosystem Services: Improving Decisions with a Systematic Approach

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Rangeland ecosystem services: improving decisions with a systematic approach

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Abstract. Delivering ecosystem services from rangelands represents a unique challenge. While social, ecological and economic complexity and diversity often lend stability to rangeland systems, the broad array of services, users and connections makes the process of identifying what services to manage for, which management practices are most effective and how to deliver them challenging. In addition, lag times between management changes and responses, climatic variability and changes in demand can further complicate decision-making. We propose a structured process that includes: (1) inventory of existing conditions; (2) identification of relevant scenarios; (3) stakeholder involvement; and (4) monitoring for verification based on the unique nature of rangelands as complex socio-ecological systems. Our objectives are to improve the quality of management planning and implementation by land managers, better inform the policies and programs that assist managers and to enhance the credibility of delivery systems. The goal of this approach is to improve sustainability by expanding the mix of ecosystem services rangelands can deliver and stabilizing income to support people who depend on rangelands.

Keywords: State-and-transition-models, socioeconomic systems, stakeholder involvement.

Introduction

Rangelands and grasslands span a vast proportion of the globe and, historically, their dominant economic use has been the extensive raising of livestock under both private and communal grazing systems (FAO 2006). While it is increasingly well recognized that these lands can generate a wide array of ecological goods and services (collectively known as ecosystem services) to humankind, the strong historical focus on the production of livestock and their produce is largely due to the fact that existing market and institutional arrangements generally encourage and reward this activity. While considerable effort is being directed to defining and quantifying the nature and level of all services provided by the 'natural capital' of range landscapes, the translation of this effort into the establishment of functional markets and institutions remains relatively modest. Without the supporting and rewarding structure of such markets and institutions, range production will remain largely skewed towards the narrower provision of grazing services and the externalities associated with under provision of other desirable ecosystem services will inevitably persist.

The relative price ratio for grazing services versus alternative non-grazing services being skewed in favor of the former is necessarily of significant concern for the ecological impact on natural resource condition. Not only can grazing conflict with the provision of many other important ecosystem services but, in the absence of profitable alternatives and in light of increased population growth, from the perspective of private range managers, an ongoing cost-price squeeze is placing considerable pressure on them to actually increase the intensity of their livestock grazing activities thereby further increasing the scope for conflict with other services and community interests.

The science and technology underlying the study of rangeland-based ecosystem services has advanced to the point that a more systematic approach is necessary to define the broad scope of ecosystem services that are potentially available from range landscape resources, allow for comparative decisions and provide certainty to markets if rangeland managers are to participate in formal markets and the public is to benefit from a better mix of service delivery. In particular, private land owners and managers require a relatively transparent approach that will allow them to reliably predict what ecosystem services can be produced from a particular tract of land under various land use and management regimes, what amounts of those services can be reliably produced, and what are the trade-offs of benefits and costs between various alternatives.

Decision-making surrounding various land management actions is typically based on reactions to a mix of historical events, current conditions and perceived future conditions. These 'filters' determine the types of actions that people will take and the success or otherwise of those actions. An inability to realistically detect and assess market signals, link those signals to changes in operations and effectively implement management responses will severely constrain the ability of rangeland managers to participate in and help create emerging markets. As a general rule, manipulating the outputs and condition of rangelands and grasslands is primarily achieved either directly or indirectly by controlling populations of grazing animals. Controlling either or both livestock and wildlife populations in time and space is difficult and usually requires substantial investments that necessarily carry degrees of risk. Three key elements in assessing the nature and magnitude of that risk are a well-developed understanding of the main financial and ecological processes, the framework of legal and social restrictions/ interactions and the inherent biophysical behaviour of rangeland ecosystems.

With reference to the first key element, how well do the individuals and institutions who develop and implement policies, programs and management actions to promote and enhance ecosystem service provision understand the basic processes that are necessary to be successful? These processes include not only the many biophysical interactions across multiple spatio-temporal scales, but also involve an ability to detect appropriate market signals sufficiently far enough in advance to redirect relatively slow responding ecosystems and operations. With respect to the complexity of information required for sound decision-making there has been substantial discussion, resources and effort devoted toward developing and organizing such information into accessible decision-making tools (Maynard *et al.* 2010, Karl *et al.* 2012). Nevertheless, the reality is that rangeland-based social-ecological systems are exceedingly complex and difficult to understand, much less direct toward precise ecosystem service outcomes (Brunson 2012). While there have been intellectually appealing and logically sound approaches proposed for developing the support systems to link management actions to ecosystem services, the challenge is real and progress will be slow (Bestelmeyer and Briske 2012). The immediate tasks are not to decide which of several competing approaches for delivering ecosystem services is best, but to develop a credible approach to selecting appropriate decision-making tools, measurement protocols and market mechanisms to insure that buyers, sellers and the public are all well served (Brown and MacLeod 2011). With respect to the timeliness of information synthesis and decision-making, the notion of 'thresholds' and 'lags' are particularly important issues for rangeland resource management. For example, these are usually thought of as points beyond which changes of states of some element may occur rapidly in response to various pressures including those imposed by management. These may also describe circumstances in which a state of inertia may be eventually reached from which recovery becomes exceedingly difficult. For example, many examples have been observed of sustained levels of overgrazing shifting range vegetation from highly productive to much less productive states (see Briske *et al.* 2011), and instances of local extinctions continuing to occur well beyond the time that grazing pressure may have been reduced (*e.g.* Woinarski *et al.* 2011). Figure 1 illustrates the lag in time between the introduction, recruitment and establishment of an invasive shrub (*Acacia nilotica*) in central Queensland. Within the first five years after the introduction of *Acacia nilotica* seed, seedlings have been established and are capable of surviving drought, grazing and fire. However, there is little effect on forage production or livestock performance. As juvenile shrubs increase in size and cover, forage production declines slowly, until ultimately forage supply is reduced and livestock performance suffers.

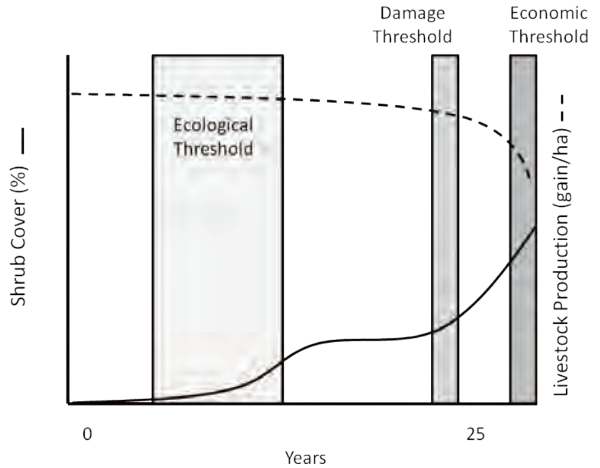

Figure 1. The interactions among shrub cover, forage production and livestock performance over a 25 year time period in the Mitchell Grasslands of central Queensland. Redrawn from Brown *et al.* 1998).

Management practices (broad-scale burning, individual juvenile treatment) to reduce the impact of shrubs are most effectively applied during the first 2-3 years, but have little direct effect on forage supply at that point and may actually require the sacrifice of some short term access to forage (Brown and Carter 1998). Whether the ecosystem service of interest is forage, meat, wool, erosion control, soil carbon stability, wildlife habitat or recreational tourism, the ecological principles and management practices for the maintenance of grassland function applies. Likewise, the time lags between the ecological threshold and economic impact threshold are largely the same.

The second key element in assessing the nature and magnitude of risk asks, what are the legal and social restrictions or incentives for the implementation of management practices and technologies that might otherwise enhance the provision of an optimal mix of ecosystem services from privately managed rangelands? For example, it is generally understood that decision makers will respond positively to strong and clear market signals and this, as noted before, is clearly the case with dominant selection of livestock production activities in most rangeland regions. It is also the basis for using additional financial incentives such a price subsidies and grants to promote wider provision of conservation activities such as clean water legislative initiatives in the USA (Clean Water Act of 1972, the Rural Clean Water Program of 1980) and Landcare and Envirofund grants in Australia. For instance, concerns about the quality of surface water in many rural and semi-rural areas have spawned the implementation of a variety of voluntary, incentive-based conservation planning approaches to encourage private landowners to adopt more sustainable management practices (Larson *et al*. 2005). Although participation in the various incentive schemes is nominally voluntary, the accompanying threat (both real and perceived) of future regulation is a powerful motivator for program participants and generally requires the implementation of a host of practices that improve ranch-level economic performance and watershed-scale wildlife habitat in addition to water quality. Well designed and implemented programs can have multiple ecosystem service benefits and provide a much higher benefit to cost return than are identified with a narrow economic analysis (George *et al.* 2011). However, and despite good intentions, some poorly designed incentive schemes can lead to seemingly perverse behavior such as clearing of native trees to make way for new tree plantations under carbon offset schemes that disallow the crediting of the sequestration capacity of existing tree stands. As a note of caution, some market signals may be both strong and well-understood, but responding to them would likely violate contemporary legal (*i.e.* endangered species habitat, air and water quality) and social (*i.e.* odor, aesthetics) restrictions. One example is provided by concessional income tax write-offs for expenditures on fuel and machinery operations that are associated land development activities that include vegetation clearing and wetland drainage with subsequent loss of habitat and biodiversity services. An example of a perverse consequence of an otherwise well-intended legislated control instrument is provided by the imposition of development restrictions on landholders under the US Endangered Species Act. While this instrument may have positive effects for wildlife generally, it can also encourage pre-emptive habitat destruction by private landowners who fear losing the use of their land because of the presence of an endangered species and may even lead to deliberate killing of some endangered species to avoid their discovery (Lueck and Michael 2003, Kroeger *et al.* 2010).

The impact of conservation incentive and regulatory schemes is of considerable importance, and in some instances, highly formalized regulatory and incentive programs may have economic impact beyond the target lands because of the high rates of adoption of prescribed land use and management practices or the abandonment of proscribed activities. For example, the Conservation Reserve Program (CRP), instituted in 1985 by the U.S. Congress, has had positive impacts on wildlife habitat and soil erosion. But, the removal of even marginal land from production of commodity crops has had substantial impact on local and regional economies (Sullivan *et al.* 2004). In the very short term, farm-related businesses (seed, fertilizer, implements, fuel) suffered, but other businesses (outdoor recreation) expanded and offset the negative impacts in the medium term. Although the economic tradeoffs are relatively well-documented for both negative and positive incentive-based conservation programs, there has yet to be a systematic and comprehensive approach that will project what the trade offs are likely to be in terms of ecosystem services (Sullivan *et al.* 2004). In developing countries, changes in land use and management practices may occur at a rate much more frequent than more stable regions (Herrick *et al.* 2012). These shifting patterns of land use, management and production goals generally occur in extremely complex spatial patterns and can dramatically affect the provision of ecosystem services beyond the readily apparent market-place of food and fibre.

Finally, the biophysical realities of rangeland ecosystems can greatly limit the kinds of ecosystem services that can be provided and the speed with which managers can actually respond to market signals. Opportunities for the remedial manipulation of rangelands and grasslands are extremely limited and invariably highly context dependent (MacLeod and Noble 2001). Inputs are primarily in the form of management of ecological processes including, for example, use of prescribed fire and seasonal application or restriction of grazing Range remediation practices that rely on fossil fuel based inputs are generally too expensive in relation to the nature and magnitude of short-term outputs in mesic areas and seldom effective in arid and semi-arid area (Noble *et al.* 1997).

Although an ability to provide some ecosystem services (heritage, cultural) may appear to be relatively simple, the physical and financial infrastructure that may be necessary to profitably do so requires considerable commitment of both time and money, especially on the part of private land managers (Pannell 2001). The extensive nature and low intensity of soil disturbance of rangeland ecosystems have encouraged private landholders, policy-makers and researchers to promote the possibility of rangeland soil carbon sequestration as a component in greenhouse gas emission reduction programs (Brown and Sampson 2009). Initial attempts to develop global, national and project scale frameworks to allow participation of rangelands in GHG offset programs have suffered from all of these challenges (deStieger 2008, Gosnell *et al.* 2011). The Chicago Climate Exchange (CCX) was formed in 2004 to test a variety of models for the voluntary reduction of GHG emissions through improved business practices or offset trading (CCX 2009). The CCX organizers wished to include as many of the possible sources of GHG emission and sequestration as possible and actively sought advisors and participants for the formation of a rangeland grazing management offset program. Although low prices and political uncertainty hampered the participation of private landholders, CCX enrolled over 3 million ha in the western U.S. Initial enthusiasm was quite high among land owners, but the challenges of developing and adhering to project level requirements for verification and validation protocols proved overly onerous for most. In particular, the need for an aggregator entity that could organize landowners, implement verification protocols to insure compliance with defined best management practices and provide credible validation of the assumptions behind the management practices and their link to soil carbon dynamics proved to be the most challenging aspect of project execution (Gosnell *et al.* 2011). In addition, the five-year contract period was far shorter than required to provide credible measurements of changes in soil carbon in response to management practices (Booker *et al.* 2013). In discussion with the aggregators, one of the major challenges in organizing projects was the lack of a comprehensive system that would allow potential participants to identify pathways from current conditions to the provisions of new ecosystem services and the management practices, timeframes and economic risk analysis associated with the changes in management (deStieguer *et al.* 2008).

Some Key Questions

How do we translate changing market signals into changing management? Market signals for some ecosystem services associated with rangelands are clear and relatively rapid. In particular, for many of the provisioning services which include production of livestock, meat and fibers, we have previously observed a very strong set of ‘signal to action’ linkages in operation across most rangelands. (The Millennium Ecosystem Assessment (2005) defined ecosystem services in four broad categories: provisioning, regulating, cultural and supporting). For some other provisioning services of rangelands, such as a repository of genetic material of use for recovery plantings in degraded pastures (Whisenant 1999) and sites for energy extraction (oil, gas, solar, wind) or supporting infrastructure (Doherty *et al*. 2011), markets and delivery networks are less developed but are emerging along with defined standards and regulatory procedures. The market signals for the other major ecosystems service categories including the so-called regulating, cultural and supporting services, are generally weaker because with few exceptions they are inherently more difficult to define, measure and value (Brown and MacLeod 2011). Nevertheless, significant attempts are being directed to establish quasi market for these services, such as the successful Bush Tender scheme now operating in southern Australia through which public land and water resource conservation agencies employ an open tender system to purchase conservation set asides (‘habitat hectares‘) of significant native vegetation from ranchers for defined periods (Whitten and Shelton 2005). However, even for the relatively tangible provisioning services, the response times between implementing management actions to achieving observable service outcomes are generally comparatively slow. For example, the time required to convert range enterprises to new operating systems (sheep to cattle), build new tourism infrastructure, organize a carbon sequestration project or improve wildlife habitat with prescribed burning, seeding and replanting is seldom less than 5 years even when climatic conditions are favorable. In fact, the suggestion has been made (Booker *et al.* 2013) that the time frame for rangeland-based ecosystem service projects should be on the order of decades rather than years.

For a particular ecosystem service, class of ecosystem services or combination of services, an obvious question becomes ‘where do we most effectively produce those services and where should we concentrate our management efforts’? Clearly, the strong emergence of landscape and regional ecology as a science has taught us that the way that landscapes are spatially arranged can have a tremendous impact on the provision of a wide variety of ecosystem services (Ludwig and Tongway 1997). Hierarchically, the fields, farms and catchments are building blocks that ultimately determine the amount of ecosystem services. In many cases, the actual amount of particular soil, vegetation or management attributes held within a landscape are eclipsed by the spatial arrangement of those attributes across the landscape. Therefore, regardless of how they are eventually identified and valued, a significant challenge for establishing effective reward and penalty schemes to enhance the provision of ecosystem services from range landscapes is being able to attribute clear linkages between the level and timing of management inputs and specific site outcomes. Private land managers will expect to be rewarded for their efforts and risks, and equity would demand that such rewards are in line with the quality of the services that are actually being provided. The challenge then is to determine the capacity of a given unit of rangeland or grassland to provide a particular ecosystem service or suite of services, identify the benefits and beneficiaries of those services, and cost-effective systems of measurement and monitoring.

A systematic approach to decision-making for ecosystem services

The complexity of integrating social, economic and ecological information into decision-making with credible outcomes demands a systematic approach to insure transparency, repeatability and post-hoc analysis. While the 4 steps we propose below are critical in terms of the chronology and relationships, there are a variety of ways of to achieve the objectives at each step. The challenge is to make sure that information is compatible in detail and timeframe. We propose these steps and procedures:

- Inventory initial conditions at a management relevant scale (*i.e.* soil, climate, vegetation, hydrology and topography)
- Through the use of graphic and mathematical models, develop scenarios of potential ecosystem services, the management required and impacts on the ecosystem functions underpinning ecosystem services.
- Using the information generated in steps 1 and 2, implement a stakeholder centered process to identify the most likely scenarios and develop ecosystem service delivery mechanisms (projects) and management practices to achieve the identified goals.
- Develop monitoring, verification and validation protocols that meet the needs of property, project, national and international level rules and regulations.

Describing spatially explicit initial conditions

The basis for this approach begins with using state and transition models unique to each ecological site (Bestelmeyer *et al.* 2004) to define the range of potential ecosystem services that a site has the capacity to generate. An ecological site is the finest scale delineation of similar land types available. For each site, a variety of current states are possible, depending on past land use, management and weather. Each state represents a unique combination of ecological processes (hydrology, energy and nutrient flows) resulting in specific soil:vegetation combinations. These particular soil:vegetation combinations result in a unique combination of ecosystem services (Havstad *et al.* 2007). Although site behavior is often determined by processes at finer scale (vegetation patches) and many ecosystem services only emerge at coarser scales, the site scale is the most cost-efficient building block for this type of analysis (Bestelmeyer *et al*. 2011). Management units (paddocks, pastures) frequently contain more than one ecological site, but the extensive nature of rangeland use and management precludes the utility of finer scale maps. A state-and-transition model unique to each site (climate, soil, geomorphic position) describes in detail the stable soil:vegetation relationships, the ecological characteristics and defining processes of each state and the changes in ecological processes and management practices necessary to change state. Figure 2 depicts the conceptual relationships from the *Acacia nilotica* increase in Figure 1

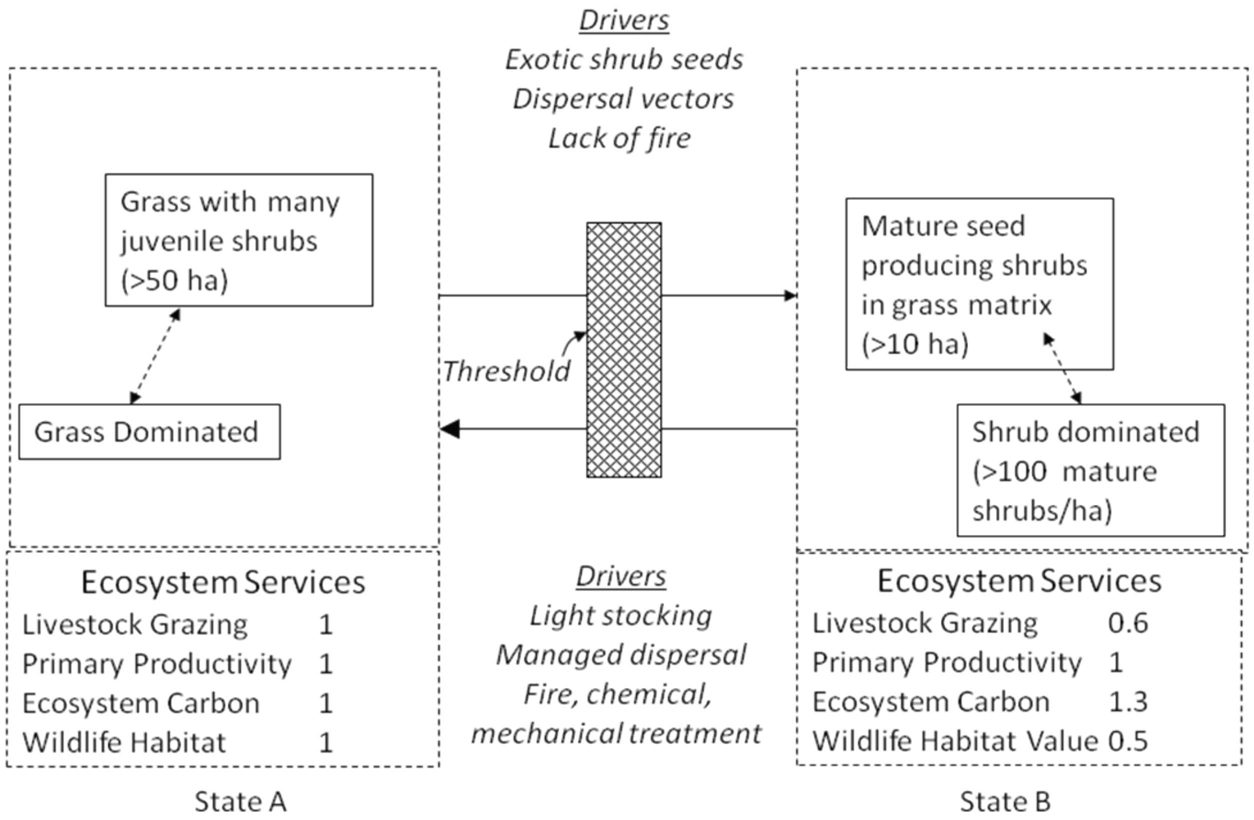

Figure 2. A graphic state and transition model for a Mitchellgrass Plains site. Ecological processes and ecosystem services derived from State A and State B are substantially different both in the expectations of products and services and in the management required to extract and maintain those services (estimates of shrub densities and ecosystem services from McArthur *et al.* 1994, Whiteman and Brown 1998 and Brown and Carter 1999).

into a more detailed state-and-transition model that also describes the changes in management and the relative level of ecosystem services for each state. Although this is a simplified example, the process for more complex sites and ecosystems is relatively well defined (Brown and MacLeod 2011). For these sites, as shrub cover increases: forage production and livestock performance decrease; soil carbon remains relatively stable, but aboveground carbon increases and ground nesting mammal habitat decreases.

From a map of ecological sites with multiple states, we can develop a range of ecosystem service maps using models with coarser scale outputs that will allow us to explore potential interactions among the basic components (Fig. 3; Brown and Carter 1998). While this example was chosen specifically because of its relative simplicity and availability of existing information, recent examples from much more complex landscapes (Steele *et al*. 2012) illustrate how a variety of changes in plant community attributes can be mapped and assessed for their ecosystem service implications, and for the management responses necessary to move from state to state.

Predicting ecosystem services under different scenarios

An extension of the state-mapping approach can also be used to make initial assessments of the time frame and management actions that are necessary to realize different types and levels of ecosystem services, the trade-offs among services for a given management action and what

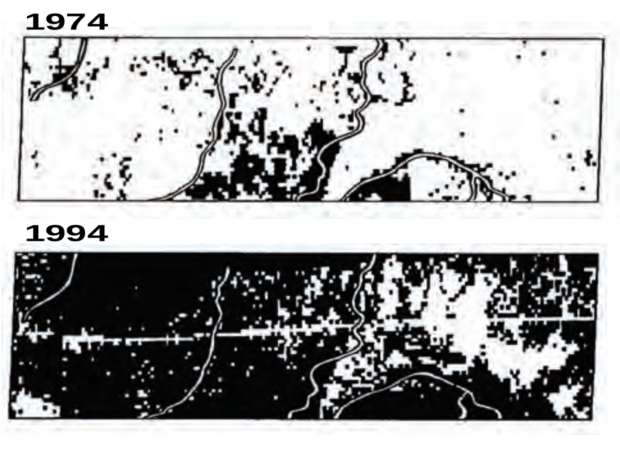

Figure 3. A map of the ecological states in a Mitchell grass landscape in 1974 and 1994. Figure from Brown and Carter 1998. State A (grassland dominated) is white, State B (Shrubland dominated) is black. Cell size is approximately 60 x 60 m.

sort of supporting, monitoring and regulating policy provisions and programs are required. Some ecosystem services are highly site specific with little influence on their provision from the land use and management at surrounding sites (*i.e*. carbon sequestration), while others may only emerge at landscape scales (*i.e.* water quality) and may depend closely on the spatial arrangements of ecological units at larger catchment or regional scales (*i.e*. wildlife habitat corridors). The spatial integration of site-specific information can be accomplished by building and integrating hierarchical models and is unique to each service and scale (Bestelmeyer *et al.* 2011, Karl *et al.* 2012).

Identifying the potential types and scope of ecosystem services that a given spatial unit (paddock, property, catchment, region) can generate presents many significant challenges, as we have already observed. Identifying actual service provision by identifying beneficiaries and placing specific monetary or social values on those services remains a greater challenge despite a growing commitment of theoretical and practical effort to the task (Murtough *et al*. 2002; MacLeod and Brown 2011). This is necessarily the case because ecosystem services often provide a mix of 'market' and 'non-market' benefits which may be further apportioned into 'use' and 'passive' values (Kroeger and Casey 2007). Related to the possession of 'market values is also the issue of many ecosystem services having attributes of *public good*' and the ability of private range managers to capture economic benefits for services they provide (*e.g*. Murtough *et al.* 2002).

Theoretically, the value of a piece of rangeland is determined by the net sum of the value of all the ecosystem services that derived from that piece of land. Some ecosystem services (*e.g.* livestock production) are relatively easy to measure (*i.e.* number of animals sold and their weight at auction) and value (*i.e.* livestock prices); some are difficult to measure (*e.g.* carbon sequestration), but easy to value (*i.e*. global market prices); some are easy to measure (*e.g*. crop genetic diversity), but difficult to value (*i.e*. cultural significance) and some are both difficult to measure and value (*e.g*. water yields). The challenge for the range and grassland research and management profession in the coming decades is to develop transparent systems for measuring rangeland ecosystem services and communicating those measures to the public, policy-makers and individual land owners and other residents. In some cases, identifying beneficiaries and the value of even precisely measured ecosystem services will be transparent, but for many others the value will always remain in the eye of the beholder and will change over time, especially where the services embody significant social or cultural values (*e.g.* landscape aesthetics, species existence values). However, to avoid problems of double counting, identifying benefits and beneficiaries is important to isolating intermediate services (*e.g.* nutrient cycling, habitat provision) to the provision of more tangible environmental products and services *(e.g*. crops, hunting opportunities: Kroeger and Casey 2007).

Not all environmental resources provide both market and non-market benefits and, regardless of joint possession of these benefit types, the relative scale of market and non-market benefits from particular tracts of rangelands will vary according to a variety of factors, including the size and richness of the local resource endowment, ecological health of the resources in situ, local land uses, adjacent land uses and opportunities for substitutes to provide similar services. One very important consideration is that while non-market benefits from natural ecosystems do present challenges for valuation, the limited available studies of broad-acre agricultural landscapes consistently suggest that these benefits may be substantial (*e.g*. Lockwood *et al.* 2000) and the continued pursuit of appropriate valuation techniques is worthwhile to promote optimal rangeland and grassland resource use.

Implementing a stakeholder-centered process

In the establishment and adoption of markets and institutions for rangeland ecosystem services, prioritization of ecosystem services occurs at two stages and scales. Each stage and scale of prioritization may require a different set of criteria corresponding to the prevailing dynamics that reflect the context of decision making, including the three key elements in assessing the nature and magnitude of risk (Nkem *et al.* 2008, Brown and Macleod 2011). To develop prioritization criteria, the first question required to be answered at any scale is, 'what is to be prioritized and for what reason?' For example, are we prioritizing the magnitude of specific individual or bundles of ecosystem services (the output of ecosystems), the state of ecosystems (the asset), or the change in output and/or state of the system?

One stage of the prioritization process occurs at the institutional scale (*e.g.* national, state, regional) which, (after responding to market signals) develop policies, plans and programs prioritizing which ecosystem services rangeland managers will be rewarded for managing or be penalized for impacting on (*e.g.* through setting land-use zones, incentive programs, standards setting, best management practice, fines). This process spatially prioritizes the landscape and defines which range managers 'can' potentially participate in markets and programs. It is important ecosystem service prioritization at the institutional scale is considered within the context of other policies and/or goals. For example, if policy goals are energy related (*e.g*. biofuel production), a different prioritization of ecosystem services (and therefore criteria) may be applied than if goals were health related (*e.g.* providing recreational opportunities to combat obesity).

Prioritization also occurs at the site scale when steps 1 and 2 of the systematic approach have been completed and the potential service provision, required management actions and impacts on ecosystem functions have been identified. This stage prioritizes which range managers 'will' potentially participate in markets and therefore which ecosystem services will be managed and where. It is argued the participation of rangeland managers in markets is based on previous involvement in incentive programs, the threat (real and perceived) of future regulation, land manager understanding of opportunity costs, risks and restoration costs associated with property management and ecosystem service provision (Brown and Macleod 2011).

Prioritization criteria for a systematic approach to priority setting need to be transparent, independent of each other and sensitive to market signals (Nkem *et al.* 2008, Brown and Macleod 2011, Maynard *et al.* 2012). The (environmental, social and economic) criteria for prioritizing ecosystem services as identified by Nkem *et al.* (2008), the European Academies Science Advisory Council (EASAC 2009), Egoh *et al.* (2010), Haines-Young (2011), Hein (2011), Luck *et al*. (2012) and Ambrose-Oji and Pagella (2012) have been synthesized, expanded on and presented in Table 1 to provide some example prioritization

Table 1. Example prioritization criteria for ecosystem services at institutional and site scales.

Example Prioritization Criteria	
Institutional Scale	Site Scale
Alignment with policies and programs at larger geo-political scales (*e.g.* Kyoto Protocol, Convention on Biological Diversity)	Sensitivity to environmental change
Currently available knowledge or evidence	Resources under particular threat from rangeland activities
Potential for capacity building	Cultural management preferences
Available expertise	Underpinning the most important economic activity (e.g. livestock production, biofuels)
Availability of broadly accepted methods for collecting and analyzing data and information on ecosystem services	Current financial position and projected position (based on scenarios)
Knowledge of the severity and time frame of management practices on expected impacts on ecosystem services	Level or risk to current profitability
Cost effectiveness of assessment, monitoring and reporting	Level of irreversible risk to current profitability
Reported success of other natural resource management (including incentive) programs	Ability to generate new knowledge of the site for better management
Landscape connectivity / networks	Scale of rangeland operation
Criticality - ecosystem services essential for existence	Do-ability (data, methods, paperwork)
Ecosystem service vulnerability/ irreversibility	Educational level of land manager
Proximity of ecosystem services to people	
Ecosystem services benefiting vulnerable communities	
Ecosystem service scarcity	
Technological substitutions	
Potential to build on other programs and policies	
Ability to scale up site scale estimates of ecosystem services to larger scales	
The opportunity to pool resources in addressing a common problem	
The economic importance of ecosystem services	
If the service is a final or intermediate ecosystem service	
Possibility to influence environmental and/or economic policy and decision making	
Resource and technical feasibility	

criteria at the institutional and site scales. Nkem *et al.* (2008, p. 18) say the 'basic requirements for prioritization are sound reasoning, competent technological and socio-economic analysis, and unbiased judgement'.

As the ecosystem services concept puts human wellbeing at the central focus of assessments, the primary goal of prioritization criteria should be to connect site scale ecological information with management actions that will enhance private and public benefiting ecosystem services that contribute to human well-being (MA 2005, Nkem *et al.* 2008). Identifying benefits and beneficiaries of each ecosystem service and not just what, for example, research scientists, federal government, land managers or industry believe are of value, are important to determining final ecosystem services, limiting double-counting, and developing policies and programs contributing positively and equitably to human well-being (Boyd and Banzhaf 2007, Nkem *et al.* 2008, Fisher *et al.* 2009, Johnston and Russell 2011, UK NEA 2011, Haines-Young 2011, Nahlik *et al.* 2012). The ecosystem services prioritized highly by these decision-makers may simply be those we know the most about, those easiest to value, those not necessarily important to others or politically biased. It is clear there is incomplete understanding of the links between the 'value' and 'importance' of ecosystem services to different people.

Integrating priorities at the institutional and site scale is essential to ensuring rangeland production can respond to (changing) market signals and institutional scale priorities; and so site scale priorities can inform and support the development of rewards and markets at the scale of institutions or ecosystem service provision. Without this exchange of information, range production will remain largely skewed towards the narrower provision of grazing services, limiting opportunities for range managers to reach their full potential, and ultimately the contribution of rangelands to maintaining or improving human well-being. Integrating priorities involves a stakeholder-centred process, an engaging and analytical process that allows for identifying the most likely scenarios, acceptable trade-offs and develops ecosystem service delivery mechanisms and management practices to achieve the identified goals (Cowling *et al.* 2008, Nkem *et al.* 2008, Maynard *et al.* 2012, Nahlik *et al.* 2012).

A stakeholder-centred process will promote dialogue and include the (differing) interests of stakeholders for a more balanced view as to the selection of criteria and which ecosystem services should receive priority attention. Determining an appropriate process to integrate priorities needs to consider the time and resources available for the actual prioritization process, including which stakeholders should be involved and how stakeholders should interact? The Millennium Ecosystem Assessment (2005) and others (Cowling *et al.* 2008, Nkem *et al.* 2008, Maynard *et al.* 2012, Nahlik *et al.* 2012) highlight many benefits of stakeholder-centred processes in developing and prioritizing information on ecosystem services, including creating a sense of ownership, transparency of process, consistency of approaches, obtaining consensus and social learning (learning while doing). There are numerous examples where participatory mapping and modelling have been applied to successfully develop decision-making tools, measurement protocols, develop scenarios, set criteria, determine tradeoffs and build consensus (IFAD 2009, Nelson *et al*. 2009, Raymond *et al*. 2009). The prioritization of ecosystem services is a dynamic process that needs to be regularly reviewed and updated as new knowledge emerges about the state of rangelands, management actions, the potential to provide ecosystem services and how this contributes to human well-being.

Designing verification and monitoring systems

A very real barrier to the widespread participation of rangelands and rangelands managers in the emerging ecosystem services markets are the lack of comprehensive protocols for measurement, monitoring and verification (Brown and MacLeod 2011). Because many of the most potentially valuable rangeland ecosystem services are not physically transported to a common market, protocols that transparently link changes in management actions to ecosystem service output are required (Maczko *et al.* 2011). Although much of the literature linking management actions to ecosystem process changes are logical, they are not yet robustly quantified (see Briske 2011). Spatially explicit description of soil:vegetation relationships are necessary to support multi-scale models of ecosystem behavior that will underlie a viable ecosystem service market.

In this paper, we have presented several important questions and challenges that must be resolved before rangelands and the humans that occupy them can fully participate in an ecosystem services market, and before human societies can fully benefit from those services. We have used examples from a variety of ecosystems, both simple and complex, and with information that has been generated over decades of study and observation for reasons other than the measurement of ecosystem services. Our goal has been to illustrate that the necessary information is mostly available, general protocols have been proposed and there are specific tools and techniques that can support landowners and managers as they make decisions about which ecosystem services to produce, how to manage sustainably for those services and how markets can credibly link buyers to sellers. The primary challenge remaining for professionals is to apply, test, evaluate and refine those tools and technologies to best serve the wide variety of producers, consumers and the markets that link them.

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